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WOOD AND OTHER RENEWABLE RESOURCES

Accounting for overfishing in life cycle assessment: new impact categories for biotic resource use

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Abstract
Purpose Overfishing is a relevant issue to include in all life cycle assessments (LCAs) involving wild caught fish, as overfishing of fish stocks clearly targets the LCA safeguard objects of natural resources and natural ecosystems. Yet no robust method for assessing overfishing has been available. We propose *lost potential yield* (LPY) as a midpoint impact category to quantify overfishing, comparing the outcome of current with target fisheries management. This category primarily reflects the impact on biotic resource availability, but also serves as a proxy for ecosystem impacts within each stock.
Methods LPY represents average lost catches owing to ongoing overfishing, assessed by simplified biomass projections covering different fishing mortality scenarios. It is based on the maximum sustainable yield concept and complemented by two alternative methods, overfishing though fishing mortality (OF) and overfishedness of biomass (OB), that are less data-demanding.
Results and discussion Characterization factors are provided for 31 European commercial fish stocks in 2010, representing 74 % of European and 7 % of global landings. However, large spatial and temporal variations were observed, requiring novel approaches for the LCA practitioner. The methodology is considered compliant with the International Reference Life Cycle Data System (ILCD) standard in most relevant aspects, although harmonization through normalization and endpoint characterization is only briefly discussed.
Conclusions Seafood LCAs including any of the three approaches can be a powerful communicative tool for the food industry, seafood certification programmes, and for fisheries management.

Keywords Life cycle impact assessment · Lost potential yield · Maximum sustainable yield · Overfishing · Seafood life cycle assessment

1 Introduction

More than 80 % of the world's fish stocks are considered fully exploited or overexploited (FAO 2012) and the global marine fish catches have stabilized around 80 million tons annually since the early 1990s (FAO 2012). However, the effort spent to catch fish has steadily increased after the catches peaked (Anticamara et al. 2011), and the fishing fleets have expanded toward deeper and more remote fishing locations (Swartz et al. 2010). Overfishing of fish stocks, which can be spatially and/or temporally separated in their reproduction, and depend on their own stock size and structure for growth, is a well-known problem (Pauly et al. 2002; Worm et al. 2009; Froese and Proelß 2010).

The present extinction rate and loss of biodiversity have been identified as humanity's most severe passing of the planetary boundaries (Rockström et al. 2009), and the millennium ecosystem assessment established overfishing as the main driver of biodiversity loss in the sea, as opposed to

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habitat change for most terrestrial systems (MEA 2005). Thus, the commercial harvesting of a few stocks indirectly affects entire ecosystems. Overfishing directly limits a biotic resource that currently accounts for 17 % of the animal protein intake worldwide, with high nutritional and economic values that are crucial for many low-income and food-deficient countries (FAO 2012).

1.1 Life cycle assessment[1]

Increased knowledge about environmental threats has raised the demand for sustainable seafood and increased the incentives to improve products and production processes (Thrane et al. 2009). Life cycle assessment (LCA) is here a useful, acknowledged, and standardized method to assess potential environmental impacts over a product life cycle from cradle to grave (ISO 2006a, b). The European Commission has concluded that LCA provides the best framework for describing the environmental impacts of products and services currently available (EC 2003) and one of the benefits is the ability to compare products and impacts in a quantitative way, either by potential impacts in terms of midpoint impact categories or by potential damage as endpoint categories. Both types of impact categories directly or indirectly target three defined areas of protection (AoPs): natural ecosystems, natural resources, or human health. It is mandatory to check and address the damage pathways towards each "relevant flow" in the inventory. If a "relevant flow" is observed, such as overfished cod and haddock when performing an LCA of fish fingers, it should be included. If fish represents a significant proportion of a product studied by LCA, e.g., fish fingers made of cod or haddock, fish is a relevant flow and overfishing should be assessed. If no impact assessment method exists to characterize the impact/damage, it should be developed and included, or clearly stated in the goal and scope definition of the LCA that it does not account for all relevant flows (ISO 2006a, b; ILCD 2010).

1.2 State of the art in seafood LCA

The theory behind biotic resource use in LCA was outlined in the 1990s and reviewed by the Society of Environmental Toxicology and Chemistry (SETAC), which led to a conclusion of a twofold impact pathway separating resource and ecosystem damage (Haes et al. 2002). The review also forecasted that more sub-impact categories would be developed to tackle the heterogeneity of impact pathways, under the broad impact category of biotic resource use (Haes et al. 2002); note that a primary production-based impact category has been proposed under the same name (Papatryphon et al. 2004).

[1] Readers with non-LCA background are encouraged to view key concept glossary in supplementary information S1a

Since the 1990s, more than 100 seafood production systems have been described with LCA, including both fisheries and aquaculture systems, the latter often depending on feed inputs from capture fisheries (Parker 2012), and a rapid increase in seafood LCAs has been recorded (Avadí and Fréon 2013). Yet none of the original methods (Haes et al. 2002) have been used in published seafood LCA case studies (Pelletier et al. 2007; Parker 2012; Vázquez-Rowe et al. 2012a; Avadí and Fréon 2013), possibly owing to lack of applicability.

The lack of methodology to assess impacts on target stocks has limited the scope of seafood LCAs, and this limited scope has been concluded to significantly impair the value for LCA as a management tool (Pelletier et al. 2007). Yet several seafood-specific impact categories have been proposed mainly regarding by-catch and discard (Ziegler et al. 2003, 2011; Emanuelsson 2008; Vázquez-Rowe et al. 2012b). Discards have also recently been characterized based on the primary production required and the amount of threatened species discarded per kilo of fish landed (Hornborg et al. 2013a, b). Some methodology for seafloor disturbance area (Ziegler et al. 2003) and a number of methods for assessing specific aquaculture impacts (Ford et al. 2012) have also been presented. Until recently, target stock sustainability has only been covered qualitatively, but restoration time was recently proposed as a metric for both single stock exploitation and primary production demand (Langlois et al. 2012), in line with a theoretical extension of the land use concept into sea use, stressing the importance of developing and incorporating the concept of marine resource use in a wider LCA concept (Langlois et al. 2011).

1.3 Principles of method development

The selection of a characterization model to quantify overfishing was based on the following three sets of criteria: (1) scientific soundness (implying compliance with ISO 14044), (2) relevance and (3) applicability. We evaluated several potential models, including primary production demand/mean trophic level, vulnerability indexes (FishBase), ecosystem models (Ecopath with Ecosim), carrying capacity dependence, historical baseline comparisons, single stock age-based population models, single stock precautionary approach limits, maximum sustainable yield (MSY) management limits, and maximum economic yield.

The theoretically most relevant model should capture both ecosystem and resource damage in a comparative way across all fish stocks and habitats. We found that all potential models were lacking either the scientific soundness or applicability for global comparisons in terms of impact pathways towards one or two AoPs. As a compromise between relevance and applicability, we developed a model based on the current goals for

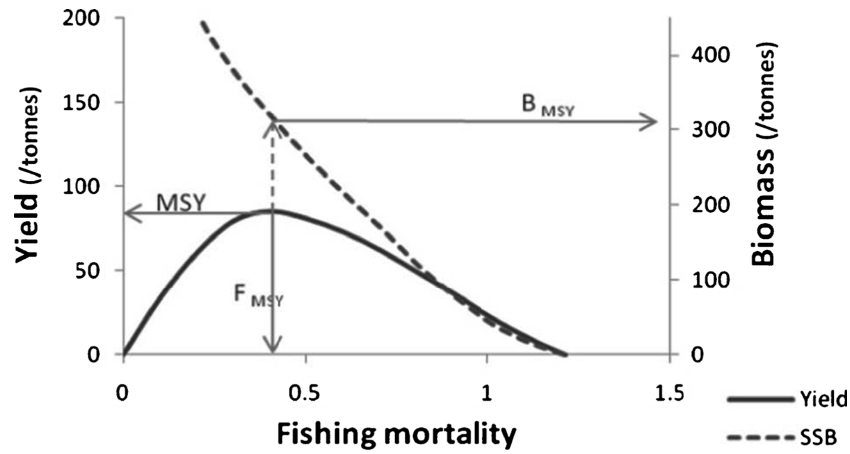


Fig. 1 MSY reference points and the relationship between fishing mortality, yield, and biomass (i.e., spawning stock biomass, *SSB*). The maximum sustainable yield (*MSY*) indicates the level of fishing mortality (F_{MSY}) resulting in a long-term biomass of B_{MSY} that could support the MSY. Reproduced with kind permission of ICES

fisheries management in the European Union, which are to follow the MSY framework, as the most practical and ISO compliant way forward; see further discussion of limitations and completeness in 4.3.

1.4 The maximum sustainable yield[2]

MSY, the theoretical maximum annual landing (or yield) that can be harvested from a wild population over time (Fig. 1), has been the backbone of fisheries science since the beginning of the twentieth century (Punt and Smith 2001). In economic terms, global fishery systems are currently far from optimized, leaving many fisheries with low profitability as a result of low stock size and overcapacity (FAO 2012). If stocks were restored to larger biomasses and after that exploited sustainably, it has been estimated that global profits would increase by US$50 billion annually, which represents more than half of the value of current landings (FAO 2008). Management according to MSY has recently been reinstated as a goal for fisheries management within the European Union, as part of the reform of the Common Fisheries Policy, which has agreed to restore all stocks to levels capable of producing maximum sustainable yield no later than 2020 (EC 2011). MSY is, however, not used as fixed goal, instead it is defined by its regulating components as follows: (1) the target fishing mortality F_{MSY} (the proportion of the stock harvested at MSY) and (2) the optimal biomass size B_{MSY} (the size of the spawning stock at MSY) (Froese and Proelß 2010; ICES 2012a).

Starting with an unfished stock as an example, increased fishing pressure over time (increasing fishing mortality F) will reduce the biomass from a pristine, i.e., unfished, condition (dotted line in Fig. 1). Note that in Europe, the biomass B in B_{MSY} typically refers to the spawning stock biomass SSB, i.e., the reproducing part of the stock (Froese and Proelß 2010). This conceptual model implies increased long-term yields with increasing fishing mortality (the solid line in Fig. 1) until $F = F_{MSY}$, after which the biomass and long-term yield will start to decrease as a result of overfishing, owing to density-dependent mechanisms. After continuous exploitation at F_{MSY}, the biomass B will fluctuate around B_{MSY}, enabling long-term average yields at MSY (Schaefer 1954; ICES 2012a). In this paper, we develop an impact category based on MSY to quantify the level of overfishing across different fish stocks, based on an anthropocentric resource-based perspective that correlates with the ecosystem damage of extracting a part of a stock.

[2] Readers with non-fisheries biologist background are encouraged to view key concept glossary in supplementary information S1b

1.5 Aim

The aim of this study was to develop a quantitative methodology to include overfishing in seafood LCAs. We suggest three midpoint impact categories for use under different conditions all based on the MSY framework, and apply them to all European fish stocks for which the needed input data is available. Except demonstrating that the suggested impact categories efficiently capture the mechanism of overfishing for these stocks, it was also a goal to analyze both spatial and temporal differences.

2 Methods

We defined the following three midpoint impact categories to account for single-stock overfishing in LCA: *lost potential yield* (LPY) and two complementary categories, *overfishing through fishing mortality* (OF) and *overfishedness of biomass* (OB). The complementary categories may be used either to interpret LPY results, or as a simpler choice when neither updated characterization factors nor input parameters are available.

In this context, we defined a stock to be fished too hard in relation to MSY, resulting in *ongoing overfishing*, if the rate F

of *exploitation* exceeds F_{MSY}. The exploitation rate should be distinguished from the state of the stock, saying that if the biomass B is found below B_{MSY}, then the stock should be considered as *overfished* in relation to MSY. We found this as the most suitable terminology for LCA purposes, since it relates to the present target for fisheries management in the European Union (F_{MSY}) and to optimal resource levels for biotic resource implementation in LCA (B_{MSY}, and indirectly MSY).

2.1 Main characterization model (LPY)

The main characterization function was based on *the difference in average annual yield between a projected optimal MSY scenario and a scenario based on the current level of fishing pressure*. The projection is regulated by fishing mortality F, which includes and aggregates not only (1) reported landings, but also when found relevant (and data is available) (2) discards of juveniles and (3) assessment of underreported and illegal catches (IUU). The projection is intended to quantify the present impacts of suboptimal exploitation patterns and enable comparisons of biotic resource use among seafood products originating from different fish stocks and years. It is not intended to forecast the future of the stock, since, for example, a constant F is highly unlikely.

The theoretical optimal (MSY) scenario was defined by setting F=0 until B reaches B_{MSY} and then harvesting at F_{MSY}. The difference between the projection sums of the optimal (Y_{opt}) and current yield (Y) scenarios is then divided by the sum of current yields; see Eq. (1).

$$CF_{x,y,T} = \frac{\Sigma_T Y_{opt} - \Sigma_T Y}{\Sigma_T Y} \tag{1}$$

The characterization factors (CF) generated from Eq. 1 represent mass units of lost yield per current yield, from stock × during year y, averaged over a time period T. Each CF was calculated from two time series of projected biomass (current and optimal), multiplied by the annual average fishing mortality. Since ICES (The International Council for the Exploration of the Sea, the body giving scientific advice to the EU regarding Northeast Atlantic stocks of fish and crayfish) communicates the instantaneous fishing mortality[3] measured on a log scale, the F had to be transformed into F_{annual}; see Eq. (2).

[3] *Instantaneous fishing mortality* (F_{inst}) is the F used and communicated most frequently in fisheries management, e.g., the one given in ICES advice, although it is less intuitive (measured on a log scale) than the *annual fishing mortality* (F_{annual}, the proportion harvested each year), which was our input data into the projection function. For example, an instantaneous fishing mortality of 0.5, 1, and 1.5 corresponds to an annual fishing mortality of 39, 63, and 78 %, respectively, of the spawning stock biomass of that stock harvested each year by the fishery.

$$Y_t \approx \widehat{F}_{annual,t} B_t = \left(1 - \exp\left(-\widehat{F}_{inst,t}\right)\right) * B_t \tag{2}$$

The biomass time series B_T was established using specific values of F and B, for each stock and year, and F_{MSY} and B_{MSY} defined for each stock; see Sect. 2.3 on input data. All inputs were inserted into a year-discrete Schaefer surplus production function (Schaefer 1954), which projects next year's biomass based on current biomass by adding growth and subtracting annual harvest (yield); see Eq. (3).

$$B_{t+1} = B_t + 2\widehat{F}_{MSY} B_t \left(1 - \frac{B_t}{2B_{MSY}}\right) - \widehat{F}_t B_t \tag{3}$$

The intrinsic growth rate (r) is substituted by $2*F_{MSY}$ and the carrying capacity (K) by $2*B_{MSY}$, which follows from the assumption of logistic growth (Schaefer 1954). We also used a 5 year moving average of B to establish an initial B_t to better comply with the idea of B_{MSY} as a long-term goal around which B should fluctuate, in line with previous recommendations to handle variability in seafood LCAs (Ramos et al. 2011). All other biomasses are iteratively generated from this value in the statistical software R (R 2012); see code in the ESM S3.

The default time perspective was set to 30 years; see the sensitivity analysis in Sect. 2.4. To avoid undesired effects, two logic rules were applied to the iteratively derived CFs as follows:

Logic rule 1 If a positive LPY value describes an underexploited stock ($F<F_{MSY}$) and ($B>B_{MSY}$), it should not be considered as lost yield, but rather as a buffer that enables initial exploitation higher than F in a long-term management plan. The "lost yield" in such cases is multiplied by (−1) and represents a potential future yield.

Logic rule 2 If a false-negative CF is found due to long break-even times, a value from a more conservative (larger) T=[10 20 30 100 500] should be used or the CF should be excluded from the dataset.

2.2 Complementary characterization models (OF and OB)

We define OF as a midpoint impact category based on the F/F_{MSY} ratio, but for LCA purposes, the characterization model has been expressed as $F/F_{MSY}-1$, so that the optimum case ($F=F_{MSY}$) equals no impact and in order to express the impact per kilo (e.g., F=0.3 and F_{MSY}=0.2 give OF=0.5 kg fished in excess to what should be fished following MSY per

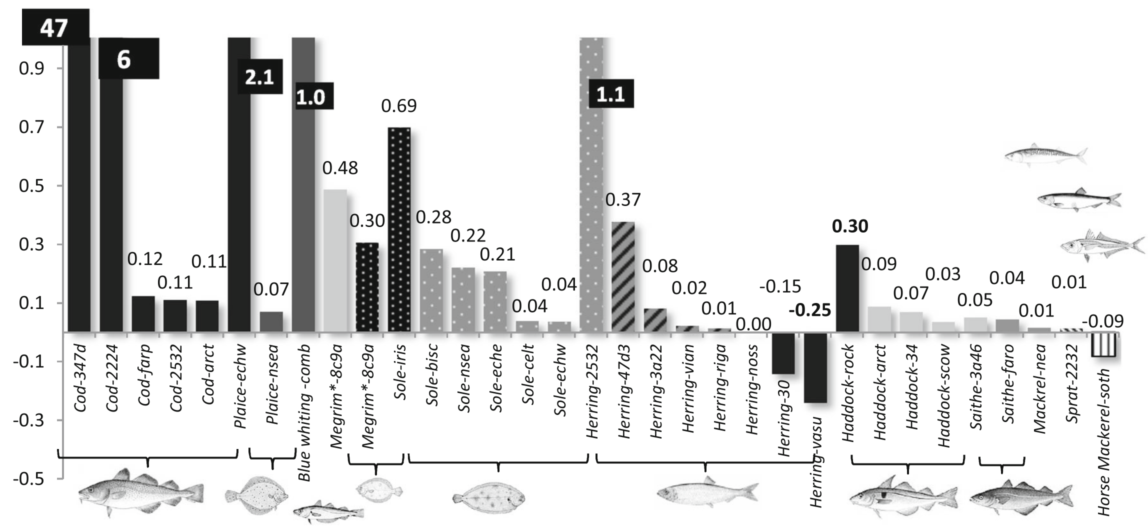


Fig. 2 LPY characterization factors for European fish stocks in 2010, sorted per species by the highest value for each stock. Note the extreme values outside the scale marked with *black boxes*, and the *negative values* describing underexploitation. For translation from stock ID to full stock names, see ESM S2. Fish illustrations used with kind permission of FAO

kilo landed). The twin category, OB, describes the present biomass state B in relation to B_{MSY}, which for LCA purposes was modeled as B_{MSY}/B-1, i.e., likewise adjusted to correspond to zero impact when B = B_{MSY}, but also inverted so that a larger value means higher impact.

2.3 Input data

The input data on fishing mortality, landings, and spawning stock biomass were retrieved from stock assessments[4] regarding the years 1995 to 2010 conducted by ICES. Data included F_{MSY} values for 31 major European stocks available in the public ICES Stock Summary/Standard Graph Database (ICES 2012b). In addition, we used corresponding B_{MSY} values from Froese and Proelß (2010). All input data are provided in the ESM S3.

2.4 Sensitivity and robustness analysis

To evaluate the model choice uncertainties, a sensitivity analysis was performed concerning two main aspects: firstly, the dependence on the time period T was tested from 0 to 500 (approximating infinity) and, secondly, the F_{MSY} values were replaced by the ones calculated by Froese and Proelß (2010). To further ensure the robustness of the LPY results, we also verified them by trends in the OF and OB impact categories, i.e., the F/F_{MSY} and B/B_{MSY} ratios throughout the time series, and we discussed qualitatively the input variability via uncertainty ranges of F, B, F_{MSY}, and B_{MSY}. General trends in temporal and spatial variability per species were analyzed by comparison of the coefficients of variation for stocks of Atlantic cod (Gadus morhua), haddock (Melanogrammus aeglefinus), herring (Clupea harengus) and sole (Solea solea), respectively, which all had more than four stocks in the dataset.

3 Results

The LPY characterization factors varied considerably between species, and even more between stocks within species; see Fig. 2 and Table 1. Three out of 31 stocks had negative LPY values, indicating underexploitation, while the remaining stocks were found to be overexploited to varying degrees. For full names corresponding to the stocks IDs given in Fig. 2, see the ESM S5.

3.1 Main results for LPY values in Europe in 2010

Generally, cod stocks had the highest LPY values, although the ranking was mostly driven by two stocks that were in an extremely poor condition in 2010: cod in the North Sea and Skagerrak (cod-3472) and Western Baltic cod (cod-2224). Plaice (*Pleuronectes platessa*) ranked second after cod, but two stocks of plaice were in highly different condition: a high value for plaice in the Western Channel (echw), and a close to median LPY value for North Sea plaice (nsea). The five species

[4] The cautious LCA practitioner will notice that both parameters B (SSB) and F do vary (slightly) retrospectively for each new stock assessment, since more data are fitted to the assessment time series, increasing the model's accuracy. For example, B regarding 2010 assessed in 2011 can be slightly changed in the 2012 assessment.

Table 1 Characterization factors of lost potential yield (LPY) for European fish stocks in 2010, based on a 20-, 30-, and 100-year time perspective

Species	ICES Stock id	$LPY_{20\ years}$ (short)	$LPY_{30\ years}$ (recommended)	$LPY_{100\ years}$ (long)	B/B_{MSY}	F/F_{MSY}
cod	cod-2224	4.33	**5.93**	10.61	**6%**	**2.3**
cod	cod-2532	0.17	**0.11**	0.04	**14%**	**0.8**
cod	cod-347d	16.1	**47.2**	263.3	**2%**	**3.6**
cod	cod-arct	0.13	**0.11**	0.07	**21%**	**0.7**
cod	cod-farp	0.16	**0.12**	0.08	**26%**	**1.3**
haddock	had-34	0.08	**0.07**	0.05	**45%**	**0.8**
haddock	had-arct	0.10	**0.09**	0.07	**86%**	**0.7**
haddock	had-rock	0.32	**0.30**	0.27	**40%**	**0.5**
haddock	had-scow	0.05	**0.03**	0.01	**35%**	**1.0**
herring	her-2532-gor	0.69	**1.05**	1.74	**22%**	**2.0**
herring	her-30	-0.17	**-0.15**	-0.10	**200%**	**0.7**
herring	her-3a22	0.10	**0.08**	0.05	**32%**	**1.2**
herring	her-47d3	0.40	**0.37**	0.34	**71%**	**0.5**
herring	her-noss	0.00	**0.00**	0.00	**141%**	**1.1**
herring	her-riga	0.00	**0.01**	0.03	**81%**	**1.2**
herring	her-vasu	-0.27	**-0.25**	-0.21	**100%**	**0.6**
herring	her-vian	0.03	**0.02**	0.01	**49%**	**1.1**
horse mackerel	hom-soth	-0.10	**-0.09**	-0.05	**139%**	**0.8**
mackerel	mac-nea	0.01	**0.01**	0.01	**91%**	**1.2**
megrim*	mgb-8c9a	0.13	**0.30**	0.84	**72%**	**1.9**
	mgw-8c9a	0.53	**0.48**	0.42	**22%**	**0.4**
plaice	ple-echw	1.31	**2.10**	6.60	**22%**	**2.4**
plaice	ple-nsea	0.11	**0.07**	0.02	**26%**	**1.0**
saithe	sai-3a46	0.05	**0.05**	0.05	**50%**	**1.3**
saithe	sai-faro	0.02	**0.04**	0.08	**72%**	**1.4**
sole	sol-bisc	0.30	**0.28**	0.23	**26%**	**1.5**
sole	sol-celt	0.05	**0.04**	0.03	**61%**	**0.8**
sole	sol-eche	0.18	**0.21**	0.24	**43%**	**1.6**
sole	sol-echw	0.05	**0.04**	0.01	**44%**	**0.9**
sole	sol-iris	0.53	**0.69**	0.64	**19%**	**1.7**
sole	sol-nsea	0.18	**0.22**	0.25	**40%**	**1.5**
sprat	spr-2232	0.01	**0.01**	0.01	**111%**	**1.2**
blue whiting	whb-comb	1.01	**1.01**	1.01	**101%**	**1.0**

CFs with LPY indicating underexploitation are highlighted in dark gray, and CFs corrected for "false negatives" due to short time perspectives are highlighted in light gray. See ESM S5 for full stock names and ESM S4 for additional time perspectives (10 and 500 years)

*The commercial fish name "Megrim" actually represents two closely related species: *Lepidorhombus whiffiagonis* (mgw-8c9a) and *Lepidorhombus boscii* (mgb-8c9a)

with the lowest LPY values, i.e., closest to the optimum level were all pelagic, lower trophic-level species, typically of smaller body size. Southern horse mackerel (*Trachurus trachurus*) and two stocks of herring were actually found to be underexploited in 2010 with negative LPY values: herring in the Bothnian part of the Baltic Sea (her-30) and Icelandic summer-spawning herring (her-vasu); see Table 1.

3.2 Temporal variation and influence of OB and OF

The temporal variation in characterization factors can be illustrated by the historical development of OB and OF regarding three stocks as follows: Eastern Baltic cod improving from significant overfishing ($F>>F_{MSY}$), North Sea plaice following a similar but less dramatic development, and Baltic Bothnian herring with negative lost yields over the whole

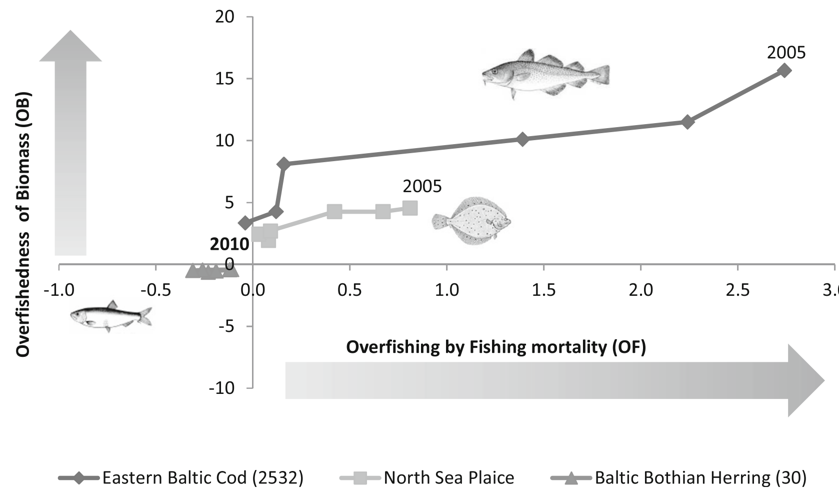


Fig. 3 Temporal variation in characterization factors of OB and OF for Eastern Baltic cod, North Sea plaice, and Baltic Bothnian herring between 2005 and 2010. Fish illustrations used with kind permission of FAO

period; see Fig. 3. The variation in LPY values over time can be seen in Table 2.

Coefficients of variation were higher between stocks of the same species (cod, haddock, sole, and herring with more than four stocks per species) than between years for each stock, indicating a larger variation between stocks than over time.

3.3 Sensitivity analysis

When time perspectives up to 500 years were tested, three groups of stocks could be observed as follows: (a) the constantly increasing, (b) the stabilizing, and (c) the stabilizing false positives (underexploited). In fact, all LPY trends are by definition stabilizing over time (owing to a constant proportion of the stock being harvested) but at different rates, see Fig. 4.

The highest stock exploitation rate (OF) and status (OB) results in a projection (LPY) that increases over a longer time perspective, as illustrated by Western Baltic cod in 2010 (cod-2224); see (1) in Fig. 4. Most projections, however, render a quicker stabilizing pattern—such as for North Sea plaice 2010 (ple-nse) (2) or for Bothnian Baltic herring (her-30) (3)—that decreases over time (note sign adjustment according to logic rule 1). Mainly based on problems observed for very short and very long time perspectives, we decided to use 30 years as the default projection time, see also discussions in 4.2, but provide projection results for longer and shorter time frames in the ESM S5.

4 Discussion

The present study provides midpoint characterization factors (CF) for LPY, OF, and OB in 2010 for 31 European stocks assessed by ICES, enabling inclusion of overfishing in seafood LCAs. The LPY is a metric of resource loss from an anthropocentric perspective, and in our view it is fully comparable between stocks in terms of resource loss, but not in terms of ecosystem damage. However, it correlates with ecosystem damage within each stock and could be used for temporal comparisons. It is important to be aware of the difficulties in describing complex biological systems with simple metrics and their limitations. We therefore encourage qualitative description of the actual ecosystem and species affected by the fishery to be included in seafood LCAs. Nevertheless, being able to quantify overfishing, even in a

Table 2 Values for OF, OB, and LPY characterization factors for Eastern Baltic cod, North Sea plaice, and Baltic Bothnian herring between 2010 and 2005

	Eastern Baltic Cod			North Sea Plaice			Baltic Bothnian Herring		
	OF	OB	LPY	OF	OB	LPY	OF	OB	LPY
2010	−0.04	3.35	0.1	0.1	1.9	0.1	−0.2	−0.6	−0.1
2009	0.12	4.26	0.1	0.0	2.4	0.1	−0.2	−0.5	−0.1
2008	0.16	8.09	0.1	0.1	2.7	0.1	−0.1	−0.4	−0.1
2007	1.39	10.11	2.5	0.4	4.3	0.2	−0.1	−0.4	−0.1
2006	2.24	11.50	9.5	0.7	4.3	0.4	−0.3	−0.4	−0.2
2005	2.74	15.67	14.4	0.8	4.6	0.5	−0.3	−0.5	−0.2

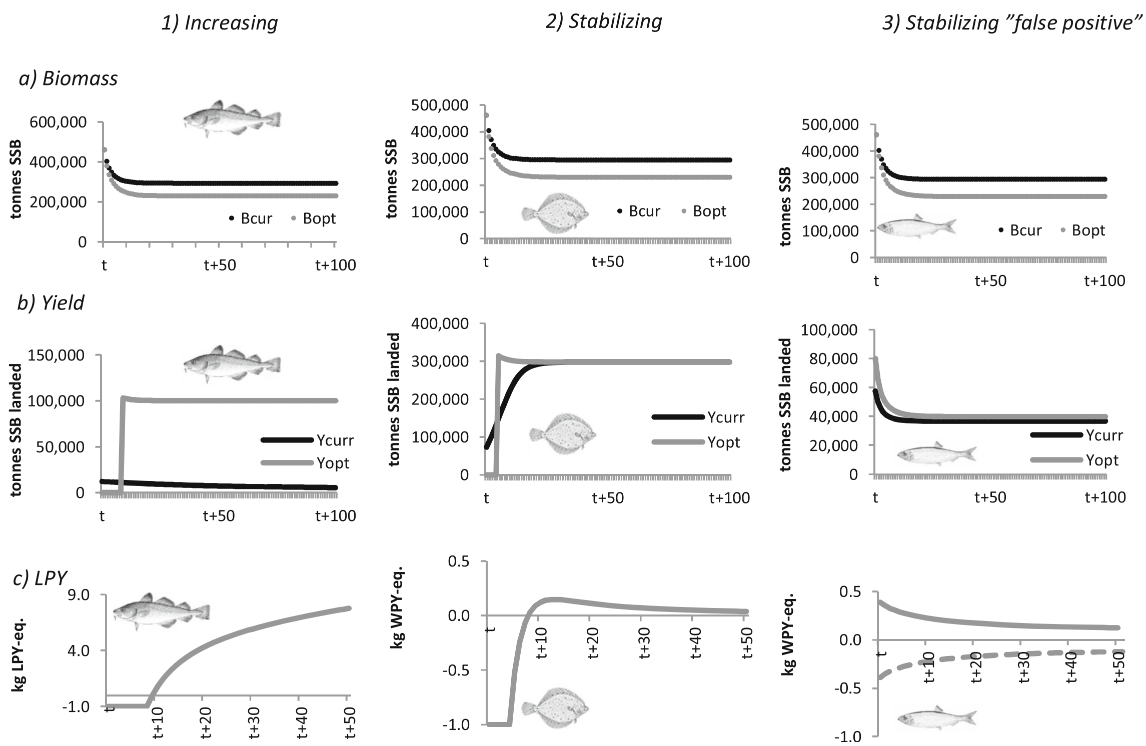


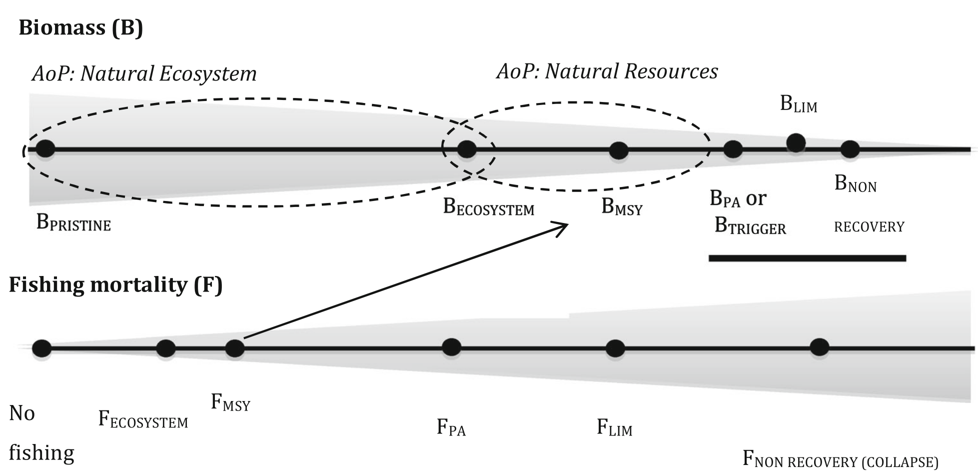
Fig. 4 Influence of increased time perspective t (all *x*-axes), plotted against the long-term development in **a** biomass, **b** yield, and **c** LPY of Western Baltic cod, North Sea plaice, and Bothnian Baltic herring. Three typical patterns are displayed: (*1*) the constantly increasing, (*2*) the stabilizing, and (*3*) the stabilizing "false positive." Fish illustrations used with kind permission of FAO

crude way, as can be done by applying the methods suggested here, represents an important step forward from not describing these impacts at all, or only describing them qualitatively.

The methodology could also be used beyond LCA as a way to rank fish stocks according to their status. Characterization through either one of the three metrics offers various possibilities to account for target stock impacts, even when all data to calculate LPY is not available.

The variation in all sets of CFs was considerable between species, but even larger between stocks within each species. Therefore, when overfishing is to be included in LCA, the stock is the necessary spatial resolution just as it is in fisheries management. The temporal analysis showed that LPY, OF, and OB values also varied substantially over time, indicating that changes in stock status resulting from natural variation and management actions are well reflected in all three CFs. The large spatial and temporal variation requires novel approaches for LCA, such as dynamic CFs that need to be updated each year for the stock(s) under study, to ensure representativity and accuracy. This annual updating could be done either by each LCA practitioner when needed or in a database to facilitate application.

Fig. 5 Schematic overview of relationships between reference points for biomass (B) and fishing mortality (F). Each fishing mortality has a corresponding long-term biomass (see example F_{MSY} and B_{MSY} marked with an *arrow*) located at different distances from the LCA areas of protection (AoPs), marked with *dotted circles*



4.1 Verification

The LPY methodology involves both model and input uncertainties on top of the natural stock variability. While the joint uncertainty is hard to assess, some aspects can be verified as follows: (1) the LPY, although not independent from them, can be compared with the more robust components OF and OB separately, (2) the development over time in all impact categories can be compared with qualitative descriptions of stock status, and (3) the LPY can be compared with other assessments of lost yields.

Fishing mortality (F) is the prime indicator used to regulate fisheries (EC 2008a), and the F/F_{MSY} ratio is widely used to evaluate MSY management (Gutiérrez et al. 2012). The parameter B_{MSY} is more uncertain, and there is considerable debate about how to calculate and apply it in practice (Agnew et al. 2013). However, Froese and Proelß (2010) provided uncertainty ranges for F_{MSY} and B_{MSY} for all stocks included in this study, showing a narrower uncertainty range of F_{MSY} than of B_{MSY}, based on the average uncertainty of three modeling approaches. For a few ICES stocks for which uncertainty ranges are provided in the assessment, such as Western Baltic cod, F also has a narrower uncertainty range than B (ICES 2011). As a consequence, the iterative characterization model of LPY magnifies this uncertainty and therefore is less robust than OB and OF, the latter being the most robust alternative.

ICES provide F_{MSY} values for an increasing proportion of the stocks assessed, but currently no information is given on B_{MSY}. Instead, a lower biomass level B_{LIM} (also called $B_{TRIGGER}$) is used as a limit above which the biomass is allowed to fluctuate, a strategy to counter the natural variation in biomass (ICES 2012a). See Fig. 5 for an overview of biological reference points for fisheries and how they relate to each other.

Another way of verifying, or at least illustrating, the LPY method would be to follow certified fisheries over time to see if certification of a previously overfished stock correlated with a drop in LPY and vice versa (suspension of certification with an increase). The Eastern Baltic cod stock was certified by both the Marine Stewardship Council and Swedish KRAV in 2010 (KRAV 2010; MSC 2013a), which correlates well with a major drop in LPY in 2009 due to a reduction in fishing mortality. The Portuguese sardine fishery, on the other hand, had its MSC certification suspended in 2010 (more recently, the suspension has been lifted), mainly because of low recruitment (MSC 2013b), but the effect on F and SSB translates into LPY values.

The concept of accounting for lost yield due to current fishing practice is not novel outside the LCA community, and it is typically based on landings (L) relative to the maximum long-term yield, MSY (FAO 2008; Froese and Proelß 2010). However, such a way to calculate lost yields in LCA has three major drawbacks. Firstly, it does not take into account the development of a stock and a heavily overfished stock could still have L = MSY, while the stock would be at high risk of collapsing. A moving average of landings could partly solve this problem. Secondly, the MSY is not itself the direct goal of fisheries management, which are rather F_{MSY} and in some cases B_{MSY}. Thirdly, the total fishing mortality is a much more accurate parameter than landings only, since it includes illegal catches and discards. The use of landing data to assess the condition of fish stocks has recently been questioned (Hilborn and Branch 2013). Therefore, in our opinion, the LPY is a more accurate characterization model than an MSY/L-based model, although the latter could represent a value for comparison or even a last choice, rather than excluding the issue entirely, if neither LPY, OF, or OB can be calculated.

In Europe, the fishing mortality of many heavily exploited gadoid stocks has been reduced during the past decade, but the biomass of many stocks has not yet been restored ($B<B_{MSY}$) (Kraak et al. 2013). The LPY values captured this development and led to lower values of lost yield than a MSY/L-1 reference calculation (see the ESM S4), since LPY responds rapidly to reduced fishing mortality. This is a desired property of an indicator for stock status used to follow up on the Marine Strategy Framework Directive (EC 2008b) and could also be attractive for fisheries certification, since, for example, the MSC can certify fisheries targeting stocks above B_{LIM} but still below B_{MSY} (Fig. 5) when stocks are moving toward B_{MSY} (Gutiérrez et al. 2012).

4.2 Model choice uncertainty

State of the art stock assessment models are age structured, i.e., taking into account the distribution of different age classes. In this sense, the Schaefer model that we used (which disregards age structure) is obsolete in terms of assessing the biomass. However, this was not what we used it for, we only used it to characterize the relationship of a number of key parameters to each other, and these key parameters are outputs of state of the art stock assessments (B, F, F_{MSY}, and B_{MSY}). The Schaefer model, which is well acknowledged for capturing the main principles of stock dynamics, thus satisfying the demand for characterization models to be "based on an distinct environmental mechanism and reproducible empirical observation" (ISO 2006b). We found that the combination of a well-known relationship (too crude for use in stock assessment) and state of the art input data represents a balanced and applicable approach for LCA characterization, as simple as possible but as complicated as necessary. The World Bank used

the Schaefer model in a global assessment of lost yields (FAO 2008) and they also used a Fox model as an alternative. Using the Fox model in our context would result in a weaker (less conservative) response to high fishing mortality and more complicated calculations (FAO 2008).

We used and recommend 30 years as the default projection time based on the motives presented previously (problems arising with very short and very long projection times and minimization of the number of occasions when logic rule 2 is applied), but we acknowledge that it could have been chosen differently, thus other sets are presented in the ESM S5. The LPY values generally increase with a larger number of iterations for all categories of F, B, F_{MSY}, and B_{MSY} combinations, however, penalizing overexploited stocks (the constantly increasing category) the most. However, the ranking between stocks remains essentially the same; thus, for decision support and quick ranking of stocks, we find the model robust with respect to the time perspective T which we tested from 0 to 500 years. Strictly, however, the time perspective T should be interpreted as an iteration number to compare stocks during 1 year, rather than as a forecast and the number chosen is of less importance as long as the same number is used throughout the LCA study.

To counter the unwanted uncertainty caused by natural variation of stocks, we used a 5-year average consistent with previous seafood application (Ramos et al 2011). The British seafood –specific carbon footprint standard PAS 2050-02 (BSI 2012) also stresses the need for average values in general but suggests 3 years. We considered 3, 5, and 10 years average but choose 5 years, where a shorter period lead to a quicker response to change, while a longer period gave more stable but less up to date values. We found 5 years to be a sound middle way given that the primary application will not be to “find the alarms” which more efficiently should be assessed with multiple parameters and local expertise, and a 10 year dataset would be too old.

The F_{MSY} values suggested by Froese and Proelß (2010) deviate from the 2012 F_{MSY} target values used by ICES, with notable consequences for seven stocks (deviating more than 50 %). In this study, we chose the ICES F_{MSY} dataset as the default one, since it is the one supported by “a large international body.” However, the availability of the independent dataset with uncertainty ranges presented by Froese and Proelß (2010) are beneficial for the methodology as a whole.

4.3 Completeness of scope

We have assessed the methodology as “compliant in most essential aspects” with the ILCD standard (ILCD 2010) and consequentially also with the ISO standards (ISO 2006a, b), even though some aspects of biological variation and complexity are previously untested in the LCA framework. The provided impact categories are on midpoint level, yet it is important to check how well they cover the relevant damage to the three defined areas of protection (AoPs) (Haes et al. 2002; ILCD 2010).

The LPY is measured in mass units of lost unspecified round weight of fish, with a clear impact pathway toward the natural resource AoP, where our definition emphasizes the role of fish as a limited and crucial protein supply (measured in unspecified mass units of lost biomass), although it easily could be further characterized into monetary units (sale price per species). In terms of damage to the natural environment AoP, LPY indirectly indicates a impact pathway towards AoP natural ecosystem. At this point we did not find it meaningful to quantify the difference between species or stocks in terms of complex ecosystem damage. However, further studies could assign various spatial and temporal weights of ecosystem vulnerability. Thus, the LPY methodology primary captures an anthropocentric resource-related aspect comparative between all stocks, with a “bonus” of ecological aspects valid only per stock (temporal comparisons).

It is worth noting that the AoPs are not precise targets in terms of biomass size, since natural ecosystem damage initially will be low or arguably neglectable due to buffering effects, while the resource AoP may vary depending on definitions (net profit, protein supply, energy) and pure assessment variability (Fig. 5). Taken together, we agree with the SETAC conclusions (Haes et al 2002) that separate methods will be needed to fully capture the damage pathways towards both of the relevant AoPs, and in some cases, even a multitude of complementary biological impact categories, as have been observed in seafood LCA development the last decade (see 1.2). However, this does not diminish the future need of some fully comparable and rougher endpoint characterization methods, as long as they are complemented with specialized quantitative midpoint categories.

The category indicators are not related to a reference unit which is common practice in LCA, e.g., like greenhouse gas emissions, which are measured in CO_2 equivalents. This is a necessity since no static reference stock exists; as an example, if LPY were measured in “North Sea Cod 2010 equivalents,” this would introduce and add the uncertainty and biological variability of the reference stock to all other CFs in the dataset. Modern stock assessment is based on time series fitting (ICES 2012a), meaning that the previous year's data will be updated and improved in each forthcoming annual assessment, making a reference stock impractical.

Without presenting any normalization scores in this publication, we note that all of the suggested impact categories are in theory possible to normalize for European waters, since approximately 74 % of European landings are covered by the 31 included stocks (representing 7 % of global marine capture fisheries).

4.4 Applicability

The LPY score could today function as a quick index to assess, rank, and communicate seafood products, and in the future, function as a stepping stone for further biological impact category development in LCA. At present, application of the suggested methodologies would probably be most useful for producers or certifying organizations and certifying organizations, either as a part of a midpoint LCA or as a stand-alone stock summary tool. Any LCA practitioner or fishery expert with a basic understanding of biological systems could retrieve and use CFs from a publication or database, or calculate new and updated CFs using the presented algorithm (see the stepwise user guide in the ESM S2). For recalculation, the practitioner has to collect F, B, F_{MSY}, and B_{MSY} values for the stock(s) and year(s) under study, for example, ICES to calculate the relevant spatially and temporally valid characterization factors as input data for an R script/spreadsheet software (ESM S3).

Depending on data availability, the practitioner may choose any combination of the fishery-specific impact categories, although LPY with OF and OB as a complement to support the discussion is preferred. Final scores should be used as a comparative index of single stock overfishing complemented with other seafood-specific impact categories or qualitative descriptions of more complex ecosystem impacts when relevant. Thus, one should acknowledge the limitations of the anthropocentrically based LPY in terms of completeness while utilizing the strengths of comparability between stock and years, as well as the relevance of using the same key parameters as used in modern fisheries management. As the temporal resolution is crucial, a practical implication for future LCA databases would be a necessity to evolve into more dynamic platforms, preferably updated on a yearly basis and complemented by a qualitative description of the stocks and ecosystems concerned. Such a development could also benefit a broader non-LCA public in condensing various sustainable fisheries data.

A central limitation of LPY is, however, that it only can be used on stocks for which the required input data are available, which in practice means only the most important commercial stocks. However, these are also the stocks most likely to be assessed in LCA studies. For other fish stocks that are affected by a fishery, either as target or by-catch species, other complementary methods will be required, examples being the recently developed fishery-specific methods described in Sect. 1.2 Seafood LCAs including the new approaches suggested here and elsewhere represent a more powerful tool for the food industry, seafood certification programmes and for fisheries management.

At present, no guideline exists for biotic impact assessment, which might result in double counting or erroneous impact mechanisms being used, if multiple ad hoc non-harmonized seafood-specific impact categories are utilized, e.g., not fulfilling the demands of ISO or ILCD. Thus, there is an urgent need for explicit guidelines to deal with biological uncertainty, which could lead to a boost of LCAs used to describe, optimize, and facilitate the path toward sustainable fisheries.

5 Conclusions

Overfishing can be quantified in terms of LPY, a midpoint impact category comparing the outcome of current versus target fisheries management. Stock and year are the optimal resolution in seafood LCAs when applying LPY, which means that a characterization factor per stock needs to be updated every year for best spatial and temporal resolution. The additional impact categories of overfishing through fishing mortality (OF) and overfishedness of biomass OB are simpler alternatives, suitable when less data is available, or to facilitate interpretation of the LPY. Characterization factors for 31 European fish stocks in 2010 are provided, and showed major variation both between stocks and over time. Seafood LCAs including any of the three approaches can be a powerful communicative tool for the food industry, seafood certification programs, and for fisheries management.

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References

Agnew DJ, Gutiérrez NL, Stern-Pirlot A, Smith ADM, Zimmermann C, Sainsbury K (2013) Rebuttal to Froese and Proelß "evaluation and legal assessment of certified seafood". Mar Policy 38:551–553

Anticamara JA, Watson R, Gelchu A, Pauly D (2011) Global fishing effort (1950–2010): trends, gaps, and implications. Fish Res 107(1–3):131–136

Avadí A, Fréon P (2013) Life cycle assessment of fisheries: a review for fisheries scientists and managers. Fish Res 143:21–38

BSI (2012) PAS2050-2:2012—assessment of life cycle greenhouse gas emissions—supplementary requirements for the application of PAS 2050:2011 to seafood and other aquatic food products. The British Standards Institution, London

EC (2003) COM 302 Integrated product policy—building on environmental life-cycle thinking. Commission of the European Communities, Brussels

EC (2008a) Council regulation No 99/2008 concerning the establishment of a community framework for the collection, management and use of data in the fisheries sector and support for scientific advice regarding the Common Fisheries Policy. Council Regulation (EC), Brussels

EC (2008b) Directive 2008/56/EC—establishing a framework for community action in the field of marine environmental policy (Marine strategy framework directive). Official Journal of the European Union, Brussels

EC (2011) COM 425 Proposal of the European Commission for a Regulation of the European Parliament and of the Council on the Common Fisheries Policy. Communication from the Commission to the Council and the European Parliament, Brussels

Emanuelsson A (2008) Bycatch and discard in Senegalese artisanal and industrial fisheries for southern pink shrimp (*Penaeus notialis*), vol 774. SIK, Gothenburg

FAO (2008) The sunken billions—the economic justification for fisheries reform. World Bank and United Nations Food and Agriculture Organization, Washington DC

FAO (2012) The state of world fisheries and aquaculture. United Nations Food and Agriculture Organization, Rome

Ford JS, Pelletier NL, Ziegler F, Scholz AJ, Tyedmers PH, Sonesson U, Kruse SA, Silverman H (2012) Proposed local ecological impact categories and indicators for life cycle assessment of aquaculture: a salmon aquaculture case study. J Ind Ecol 16(2):254–265

Froese R, Proelß A (2010) Rebuilding fish stocks no later than 2015: will Europe meet the deadline? Fish Fish 11(2):194–202

Gutiérrez NL, Valencia SR, Branch TA, Agnew DJ, Baum JK, Bianchi PL, Cornejo-Donoso J, Costello C, Defeo O, Essington TE, Hilborn R, Hoggarth DD, Larsen AE, Ninnes C, Sainsbury K, Selden RL, Sistla S, Smith ADM, Stern-Pirlot A, Teck SJ, Thorson JT, Williams NE (2012) Eco-label conveys reliable information on fish stock health to seafood consumers. Plos One 7(8):e43765

Haes HAU, Finnveden G, Goedkoop M, Hofstetter P, Jolliet O, Klöpffer W, Krewit W, Lindeijer E, Muller-Wenk R, Olsen SI, Pennington DW, Potting J, Steen B (2002) Life cycle impact assessment: striving towards best practice. Society of Environmental Toxicology and Chemistry (SETAC), Brussels

Hilborn R, Branch TA (2013) Fisheries: does catch reflect abundance? No, it is misleading. Nature 494(7437):303–306

Hornborg S, Nilsson P, Valentinsson D, Ziegler F (2012) Integrated environmental assessment of fisheries management: Swedish nephrops trawl fisheries evaluated using a life cycle approach. Mar Policy 36(6):1193–1201

Hornborg S, Belgrano A, Bartolino V, Valentinsson D, Ziegler F (2013a) Trophic indicators in fisheries: a call for re-evaluation. Biol Lett 9(1):20121050

Hornborg S, Svensson M, Nilsson P, Ziegler F (2013b) The IUCN Red List as a framework for communication regarding discard impacts in fisheries. Env Mgmt 52(5):1239-1248

ICES (2011) Report of the Baltic Fisheries Assessment Working Group (WGBFAS) : ANNEX: WGBFAS Cod in 22–24, Copenhagen

ICES (2012a) ICES Advice 2012 Book 1–Introduction, overviews, and special requests. The International Council for Exploration of the Seas, Copenhagen

ICES (2012b) Stock summary database, at ICES website. The International Council for Exploration of the Seas. Accessed 1 December 2012

ILCD (2010) ILCD handbook—general guide to life cycle assessments—detailed guidance, 1st edn. Joint Research Center of the European Commission, Luxenburg

ISO (2006a) ISO14040 Environmental management—life cycle assessment—principals and framework. International Organization for Standardization, Geneva, Switzerland

ISO (2006b) ISO14044 Environmental management—life cycle assessment—requirements and guidelines. International Organization for Standardization, Geneva, Switzerland

Kraak SBM, Bailey N, Cardinale M, Darby C, De Oliveira JAA, Eero M, Graham N, Holmes S, Jakobsen T, Kempf A, Kirkegaard E, Powell J, Scott RD, Simmonds EJ, Ulrich C, Vanhee W, Vinther M (2013) Lessons for fisheries management from the EU cod recovery plan. Mar Policy 37:200–213

KRAV (2010) Godkännade av kustnära fiske med bur, krok och garn på torsk i Östersjön. KRAV. http://krav.se/Upload/327/Beslut_torsk_ostersjon_2010-12.pdf. Accessed 2013-03-27

Langlois J, Hélias A, Delgenes J-P, Steyer J-P (2011) Review on land use considerations in life cycle assessment: methodological perspectives for marine ecosystems. In: Finkbeiner M (ed) Towards Life Cycle Sustainability Management, Chapter 9. Springer, Dordrecht, pp 85–96

Langlois J, Fréon P, Delgenes J-P, Steyer J-P, Hélias A (2012) Biotic resources extraction impact assessment in LCA of fisheries. In: Corson MS, van der Werf HMG (eds) Proceedings of the 8th international conference on life cycle assessment in the agri-food sector (LCA food 2012), 1–4 October 2012, Saint Malo, France. INRA, Rennes, France, pp 517–523

MEA (2005) Ecosystems and human well-being: Biodiversity synthesis. Millennium Ecosystem Assessment (MEA), Washington DC

MSC (2013a) Fiskbranschens Sweden eastern Baltic cod. Marine Stewardship Council. http://www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/sweden-eastern-baltic-cod. Accessed 2013-03-27

MSC (2013b) Portugal sardine purse seine. http://www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/portugal-sardine-purse-seine. Accessed 2013-03-27

Papatryphon E, Petit J, Kaushik SJ, van der Werf HMG (2004) Environmental impact assessment of salmonid feeds using life cycle assessment (LCA). Ambio 33(6):316–323

Parker R (2012) Review of life cycle assessments research on products derived from fisheries and aquaculture: a review for seafish as part of the collective action to address greenhouse gas emissions in seafood. Sea Fish Industry Authority, Edinburgh, UK

Pauly D, Christensen V, Guenette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D (2002) Towards sustainability in world fisheries. Nature 418(6898):689–695

Pelletier N, Ayer N, Tyedmers P, Kruse S, Flysjo A, Robillard G, Ziegler F, Scholz A, Sonesson U (2007) Impact categories for life cycle assessment research of seafood production systems: review and prospectus. Int J Life Cycle Assess 12(6):414–421

Punt A, Smith ADM (2001) The gospel of maximum sustainable yield in fisheries management: birth, crucifixion, and reincarnation. In: Reynolds JD, Mace GM, Redford KH, Robinsson HG (eds) Conservation of exploited species. Cambridge University Press, Cambridge UK, pp 41–66

R (2012) R: A language and environment for statistical computing. R Development Core Team. http://www.r-project.org/

Ramos S, Vázquez-Rowe I, Artetxe I, Moreira M, Feijoo G, Zufía J (2011) Environmental assessment of the Atlantic mackerel (*Scomber scombrus*) season in the Basque country. Increasing the timeline delimitation in fishery LCA studies. Int J Life Cycle Assess 16(7):599–610

Rockström J, Steffen W, Noone K, Persson A, Chapin FS, Lambin EF, Lenton TM, Scheffer M, Folke C, Schellnhuber HJ, Nykvist B, de Wit CA, Hughes T, van der Leeuw S, Rodhe H, Sorlin S, Snyder PK, Costanza R, Svedin U, Falkenmark M, Karlberg L, Corell RW, Fabry VJ, Hansen J, Walker B, Liverman D, Richardson K, Crutzen P, Foley JA (2009) A safe operating space for humanity. Nature 461(7263):472–475

Schaefer M (1954) Some aspects of the dynamics of populations important to the management of the commercial Marine fisheries. Bulletin of the Inter-American Tropical Tuna Commission 1(2):27–56

Swartz W, Sala E, Tracey S, Watson R, Pauly D (2010) The spatial expansion and ecological footprint of fisheries (1950 to present). Plos One 5(12):e15143

Thrane M, Ziegler F, Sonesson U (2009) Eco-labeling of wild-caught seafood products. J Clean Prod 17(3):416–423

Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G (2012a) Best practices in life cycle assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci Tech 28(2):116–131

Vázquez-Rowe I, Moreira M, Feijoo G (2012b) Inclusion of discard assessment indicators in fisheries life cycle assessment studies. Expanding the use of fishery-specific impact categories. Int J Life Cycle Assess 17(5):535–549

Worm B, Hilborn R, Baum JK, Branch TA, Collie JS, Costello C, Fogarty MJ, Fulton EA, Hutchings JA, Jennings S, Jensen OP, Lotze HK, Mace PM, McClanahan TR, Minto C, Palumbi SR, Parma AM, Ricard D, Rosenberg AA, Watson R, Zeller D (2009) Rebuilding global fisheries. Science 325(5940):578–585

Ziegler F, Nilsson P, Mattsson B, Walther Y (2003) Life cycle assessment of frozen cod filets including fishery-specific environmental impacts. Int J Life Cycle Assess 8(1):39–47

Ziegler F, Emanuelsson A, Eichelsheim JL, Flysjö A, Ndiaye V, Thrane M (2011) Extended life cycle assessment of southern pink shrimp products originating in Senegalese artisanal and industrial fisheries for export to Europe. J Ind Ecol 15(4):527–538